USE OF A WETLAND INDEX TO EVALUATE CHANGES IN RIPARIAN VEGETATION AFTER LIVESTOCK EXCLUSION[1]

Marc C. Coles-Ritchie, David W. Roberts, Jeffrey L. Kershner, and Richard C. Henderson[2]

ABSTRACT: A method was developed to characterize ecological integrity of riparian sites based on the abundance of hydric species. This wetland index can be calculated with species data, or with community type data as performed here. Classified riparian community types were used to describe vegetation at 14 livestock exclosures and adjacent grazed areas. Community type wetland index values were generated and used to calculate site wetland index values. It was hypothesized that removal of livestock would result in higher wetland index values because of release from herbivory and decreased physical disturbance of vegetation, streambanks, and soil. The wetland index for exclosures was about 12% higher than grazed sites; differences were statistically significant ($p < 0.01$) based on paired t-tests. The increase in hydric vegetation after livestock exclusion may have contributed to the greater bank stability ($p = 0.002$) and smaller width-to-depth ratio ($p = 0.005$) in exclosures. Challenges were encountered in using community types to describe and compare site vegetation, which could be avoided with species data collection. The wetland index can be a tool to monitor sites over time, compare sites with similar environments, or compare sites for which environmental differences can be accounted.

(KEY TERMS: riparian; ecology; monitoring; wetland indicator; livestock exclosures.)

INTRODUCTION

Streams and riparian areas perform many important ecological and social functions. Vegetation in riparian areas helps stabilize streambanks, dissipate energy of floods, support perennial flows, trap sediment, and moderate stream temperature (Gregory *et al.*, 1991; Elmore and Kauffman, 1994; Gurnell, 1997; Tabacchi *et al.*, 1998, 2000). Many of these functions are important for maintaining fish and wildlife habitat, as well as other ecological and social benefits [see reviews of Kauffman and Krueger (1984); Fitch and Adams (1998); Naiman and Decamps (1997)]. Riparian areas are a relatively small part of the landscape, yet, they support unique vegetation and provide habitat for many animals [see review of Lohman (2004)], as well as providing a variety of other benefits (such as flood abatement and recreation). As the importance of functioning riparian

ecosystems in the western United States (U.S.) has become more widely recognized, this has posed critical management issues for government agencies including the U.S. Forest Service and the U.S. Bureau of Land Management (National Research Council, 2002).

In particular, riparian vegetation and streambank condition have been linked to the health of anadromous fish runs [see reviews of Platts (1991); Fitch and Adams (1998)]. Specific concerns for fish populations within the Columbia River basin are documented in the Pacific Anadromous Fish (PACFISH, 1994), and Inland Fish (INFISH, 1995) strategies, and include maintenance of water temperature, habitat complexity, pools, sediment levels, and instream structure. The PACFISH (1994) report also notes that the quality of fish habitat is inseparably related to the integrity of riparian areas. Concern over the effectiveness of the PACFISH and INFISH strategies was expressed in Biological Opinions by the U.S. Fish and Wildlife Service (USDIFWS, 1998) and led to the establishment of the PACFISH/INFISH Biological Opinions (PIBO) effectiveness monitoring program. The objective of PIBO is to determine whether management practices in the upper Columbia River basin are maintaining or restoring the structure and function of riparian and aquatic habitats (Kershner *et al.*, 2004a).

Thus, monitoring is essential to assess the impacts of human activities on ecological functioning and the effectiveness of changes in management. Tools are necessary to evaluate changes in riparian areas and to interpret the ecological meaning of those changes. A riparian monitoring protocol by Winward (2000) was used by PIBO to describe the vegetation at riparian sites. These methods involve recording the extent of vegetation community types along the stream (greenline) and along riparian transects. A community type is an "aggregation of all plant communities with similar structure and floristic composition" (Tart *et al.*,

2002). Community types describe typical species abundances based on observed communities. A problem in using community types across a large geographical area was that multiple vegetation classifications had to be used, each of which had different names for community types regardless of how similar or different they were. This made it difficult to compare sites described by different classifications (Coles-Ritchie, 2005).

Ratings (bank stability and greenline successional status) for many riparian community types have been developed by Winward (2000). These ratings were designed to calculate a weighted average rating for the site, based on the relative abundance of each community type. This approach had several limitations. First, the PIBO study area includes eastern Washington, eastern Oregon, Idaho, northeastern Nevada, and northwestern Montana but ratings were developed only for community types of southern Idaho and Utah. Second, the ratings were not based on any defined criteria (a problem in itself), so ratings could not be consistently developed for community types in other geographical areas. Third, the ratings applied only to streambank vegetation and not to vegetation away from the stream edge, across the riparian area.

In order to evaluate all of our site community type data in a consistent and meaningful way, we developed a wetland index for riparian community types and sites. This wetland index is based on the wetland indicator status (WIS) of plant species (Reed, 1996). The index summarizes the relative abundance of species, or community types, at a site in relation to the continuum of the WIS, from obligate wetland species to upland species (Table 1). This index is an adaptation of a commonly used technique to classify riparian wetland vegetation described by Wentworth *et al.* (1988).

Evaluation of vegetation using the WIS is informative because riparian areas frequently support

TABLE 1. The Wetland Indicator Status of Reed (1996) and the Synthetic Wetland Indicator Value Assigned to Each Status.

Wetland Indicator Status	Estimated Probability a Species Will Occur in a Wetland	Wetland Indicator Value
Obligate	Almost always (99%)	100
Obligate −		92
Facultative wet +		83
Facultative wet	Usually (67-99%)	75
Facultative wet −		67
Facultative +		58
Facultative	Sometimes (34-66%)	50
Facultative −		42
Facultative upland+		33
Facultative upland	Not usually (1-33%)	25
Facultative upland −		17
Upland +		8
Upland	Almost never (1%)	1

TABLE 2. Riparian Vegetation Classifications Used to Describe Site Vegetation.

Geographic Area	Primary Classifications*	Secondary Classifications**
Northern Idaho	Hansen *et al.* (1995)	Padgett *et al.* (1989); Crowe and Clausnitzer (1997)
Central and Southern Idaho	Padgett *et al.* (1989)	Manning and Padgett (1995); Crowe and Clausnitzer (1997)
Oregon	Kovalchik (1987); Crowe and Clausnitzer (1997)	

*The primary classification used to describe vegetation at a site.
**Classifications from nearby areas that were used when the primary classification did not adequately describe the vegetation at the site.

vegetation different from the surrounding uplands (National Research Council, 2002). Riparian areas have abundant moisture because of high water table and the interaction of the channel and the floodplain through infiltration, overbank flow, and bank storage (Gebhardt *et al.*, 1989; Hughes, 1997). This creates a steep moisture gradient from the uplands to the stream edge, particularly in semi-arid regions of western North America (Castelli *et al.*, 2000; Chapin *et al.*, 2002; Dwire *et al.*, 2004). For low-gradient streams, functioning riparian areas typically have a high proportion of hydric or wetland species (see riparian vegetation classifications listed in Table 2). Disturbance in the riparian area can cause bank erosion and channel widening, which decreases connectivity (sensu Toledo and Kauffman, 2001) between the stream and floodplain. Severely reduced connectivity between the stream and riparian area results in "entrapment" of the stream within the channel such that high flows no longer reach the former floodplain (Sarr, 2002). This reduces moisture availability to plants, which can lead to a shift from obligate to facultative or upland species adjacent to the stream (Toledo and Kauffman, 2001). Such loss of riparian habitat has been shown to have negative impacts for birds (Saab *et al.*, 1995), wildlife (Lohman, 2004), and fish (Platts, 1991).

Excessive livestock grazing, which is well documented to have deleterious effects on riparian vegetation and instream condition (Belsky *et al.*, 1999), is a major source of disturbance in the study region. Rest from grazing in riparian areas can lead to significant changes in riparian and stream condition. Reduction or elimination of livestock grazing has been associated with increased streambank stability (Myers and Swanson, 1995), decreased soil compaction and increased infiltration (Bohn and Buckhouse, 1985; Wheeler *et al.*, 2002; Kauffman *et al.*, 2004), and changes in species composition (Leege *et al.*, 1981; Dobkin *et al.*, 1998) with increases in deciduous woody riparian vegetation common (Schultz and Leininger, 1990; Green and Kauffman, 1995; Case and Kauffman, 1997; Brookshire *et al.*, 2002).

In this study, the wetland index was used to compare the abundance of wetland species in riparian areas within livestock exclosures to adjacent grazed areas. We hypothesized that sites protected from grazing (exclosures) would have higher wetland index values than adjacent grazed areas. Differences in the wetland index would suggest that livestock activities have altered the vegetation or the local environmental conditions that determine which species occupy a site. We also evaluated whether changes in the wetland index were associated with changes in stream channel attributes.

METHODS

Study Sites

The data presented here are from the early years (1999-2001) of the PIBO program. PIBO uses a 5-year, alternating, rotating panel sampling design to monitor streams and riparian areas on federal land (Forest Service or BLM) within the upper Columbia River basin (Kershner *et al.*, 2004a). The stream and riparian data are collected at "response reaches" which are low-gradient (<3%) stream reaches that are sensitive to disturbances such as those caused by human activities (Montgomery and McDonald, 2002).

In this article, a small subset of PIBO sites, those with livestock exclosures, were evaluated (*n* = 14). Ten of the exclosure sites examined in this study were on Forest Service and BLM land in Idaho (Boise, Custer, Gem, Idaho, and Valley counties), and four were on Forest Service land in eastern Oregon (Grant, Harney, Union, and Wallowa counties). Exclosure fences have been used to prevent livestock from accessing parts of the stream and riparian area in order to promote maintenance or restoration of ecosystem functioning (Sarr, 2002). The average age of exclosures in this study was 13 years, based on estimates from BLM and Forest Service personnel, although the efficacy of the exclosures is not known. Exclosure sizes were approximately 0.5-5 ha (1.2-12.4 acres). The exclosure sites were all located on reaches of small, perennial streams in unconstrained valleys. Stream gradients were low, ranging from 0.1% to 2.2%, and bankfull widths were between 1 and 6 m.

Data Collection

All sites were sampled during the 1999, 2000, or 2001 field seasons (June-September). At the beginning of each field season five to eight technicians received 8 days of training in identification of dominant riparian plant species, how to use community type classifications, and the methods of sampling. Another five to eight pairs of technicians received 8 days of training in sampling physical characteristics of the stream (Kershner *et al.*, 2004a).

The monitoring methods of Winward (2000) were used to collect data about vegetation along the stream (greenline) and in the larger riparian area. The greenline is the first area of perennial vegetation adjacent to the stream that is at least one foot wide (Winward, 2000). This interface of the stream and terrestrial ecosystems is an important location to monitor because that is where disturbance can exacerbate bank erosion. In addition, the greenline vegetation reflects the level of moisture available and hence the connectivity between the stream or ground water and streamside vegetation, as described above. The vegetation of riparian transects (perpendicular to the valley direction) provides information about the larger riparian area that is influenced by the stream and the water table.

The methods of Winward (2000) use community types to describe vegetation of the riparian area, along a 110-m length of stream. Geographically appropriate vegetation classifications were used to determine the community types present at each stream site (Table 2). The extent of each community type was recorded as the data collector walked along the greenline and along five riparian transects spaced evenly along the 110 m reach. The riparian transects were all parallel and extended in both directions from the greenline to the edge of the riparian area or 27.5 m, whichever was encountered first. Botany technicians recorded the extent of each community type, using the number of steps to quantify the distance. The total number of steps for each community type along the greenline and the riparian transects were summed separately for the site, and then converted to percentages based on the total length, in steps, of the greenline or riparian transects.

Observer variability in determining the community type at a given step has been found to be relatively high, although observer variability for the wetland index generated from community type data (described below) was found to be relatively low (Coles-Ritchie *et al.*, 2004). For 11 of the 14 sites, the same technician collected data for both the exclosure and the adjacent grazed site, reducing the influence of observer variability.

Physical measurements of the stream were also collected at these 14 sites by other technicians. The length of stream sampled was a distance of 20 bankfull widths, with a minimum of 80 m and a maximum of 280 m. Channel dimensions (width, width-to-depth, entrenchment, and stream gradient) were calculated with data from four riffles. Stream gradient was measured with a surveryor's level, a tripod, and stadia rod. Bank characteristics (bank stability, bank angle, and undercuts as a percent of reach) were measured at 20 stream transects. Bank stability was evaluated based on a modified version of a protocol described by Bauer and Burton (1993) and Overton *et al.* (1997). These measures of channel and bank characteristics were found to provide relatively precise estimates in tests of multiple observers at the same sites (Roper *et al.*, 2002; Archer *et al.*, 2004).

Calculating the Wetland Index

Wetland vegetation is often evaluated or delineated in relation to WIS, using a weighted average score ranging from 1 (wetland) to 5 (upland) [for examples, see Michener (1983); Wentworth *et al.* (1988); Wakeley and Lichvar (1997); Brown (1999)]. Regional WIS for many species are defined in the U.S. Fish and Wildlife Service "1996 National List of Vascular Plant Species that Occur in Wetlands" (Reed, 1996). The WIS ranges from obligate wetland species, which are found in wetlands 99% of the time, to upland species, which are found in wetlands only 1% of the time, with 11 classes distinguished in between (Table 1). The WIS is based on the field observations and best professional judgment of scientists and the designations "represent the probability of a species occurring in wetlands *vs.* nonwetlands in the region" (Reed, 1996).

Recently, riparian vegetation has also been evaluated with the WIS by calculating the percentage of species in WIS classes (Toledo and Kauffman, 2001; Chapin *et al.*, 2002) or using the weighted average score from 1 to 5 (Stromberg, 2001). To develop the wetland index in this study, weighted average scores were calculated based on the WIS classes. As recommended by Michener (1983) and Brown (1999), Wetland Indicator Values (WIVs) were assigned to each WIS class (Table 1) by evenly distributing the values between 1 (for the upland class) and 100 (obligate wetland class). For the obligate, facultative wet, facultative, facultative upland, and upland classes, the WIVs closely correspond to midpoint of their probability of occurrence in a wetland as described by Reed (1996).

The wetland index for a site can be calculated as a weighted average of the wetland index for each species recorded at a site. For this study, community

types were used instead of species. This required calculation of a community type wetland index (CTWI). To do this, data were compiled from the published classifications (Table 2) on the species composition of each community type. Within a community type, the importance value was generated for each species by multiplying what the classification reported as the average cover of the species (average cover in plots in which the species occurred) by the constancy (percentage of plots in which the species occurred). Relative importance for a species in the community type (RImp) was calculated by dividing the importance value of the species by the sum of importance values for all species in the community type.

A CTWI was calculated as

$$\textbf{CTWI} = \sum_{i=1}^{n}(\text{RImp}_i * \text{WIV}_i) \qquad (1)$$

where RImp_I is the relative importance of species i in the community type, WIV_i (wetland indicator value) is a value from 1 to 100 for species i based on the WIS (Table 1), and n is the number of species in the community type. The maximum possible CTWI value is 100, indicating all obligate wetland species in the community type, and the lowest possible value is 1, indicating all upland species in the community type. An example calculation of a CTWI for a common *Carex* community type is shown in Table 3.

The relative cover of each community type at a site (RCov_j) was calculated by dividing the length in steps covered by each community type by the total length (total steps) sampled at the site. The site wetland index, for greenline or riparian data, was calculated as

$$\textbf{Site Wetland Index} = \sum_{j=1}^{n}(\text{RCov}_j * \text{CTWI}_j) \qquad (2)$$

where RCov_j is the relative cover of community type j at the site, CTWI_j is the community type wetland index from Equation (1), and n is the number of community types at the site.

The site wetland index can also be calculated with species data, as PIBO now does. To do so, the equation above is used, except that "j" refers to species instead of community types and the WIV value for each species is taken directly from Table 1 instead of using the CTWI.

Data Analysis

A one-sided paired t-test, with a 95% confidence level, was used to test if the wetland index values were statistically higher ($p \leq 0.05$) in exclosures compared with adjacent grazed areas. It was hypothesized that the wetland index would be higher in the

TABLE 3. Example of Community Type Wetland Index Calculation for the
Carex rostrata (now *Carex utriculata*) Community Type of Padgett *et al.* (1989).

Species	Constancy (%)	Average Cover (%)	Importance (%)	RImp[1] (%)	WIS[2]	WIV[3]	Species Score
Carex utriculata	100	89	89.0	86.7	OBL	100	86.7
Agrostis stolonifera	11	25	2.8	2.7	FACW	75	2.0
Carex nebrascensis	22	11	2.4	2.4	OBL	100	2.4
Poa pratensis	11	17	1.9	1.8	FAC	50	0.9
Glyceria spp.[4]	11	13	1.4	1.4	OBL	100	1.4
Carex aquatilis	22	6	1.3	1.3	OBL	100	1.3
Calamagrostis canadensis	11	8	0.9	0.9	FACW+	83	0.7
Deschampsia cespitosa	17	4	0.7	0.7	FACW	75	0.5
Equisetum arvense	17	4	0.7	0.7	FAC	50	0.3
Salix boothii	6	10	0.6	0.6	FACW+	83	0.5
Caltha leptosepala	6	6	0.4	0.4	OBL	100	0.4
Mentha arvensis	6	5	0.3	0.3	FACW−	67	0.2
Juncus balticus	11	2	0.2	0.2	FACW+	83	0.2
Dasiphora floribunda	6	3	0.2	0.2	FAC−	42	0.1
Trifolium spp.[5]	6	2	0.1	NA	NA	NA	NA
Cirsium spp.[5]	6	1	0.1	NA	NA	NA	NA
							97.4

[1]RImp = relative importance.
[2]WIS = wetland indicator status.
[3]WIV = wetland indicator value.
[4]Included in calculations because all species in this genus (in our study area) are OBL.
[5]Not included in calculations because species in this genus have variable WIS.

exclosures where presumably there was less disturbance of vegetation and streambanks.

Channel dimensions (bankfull width, width-to-depth, entrenchment, and stream gradient) and streambank characteristics (stability, angle, and undercut) were also compared for the pairs of exclosure and grazed sites because they have been shown to respond to changes in livestock management (Kauffman *et al.*, 1983; Magilligan and McDowell, 1997). These variables also influence the amount of moisture available to plants, and hence the abundance of obligate wetland species (Toledo and Kauffman, 2001).

Linear regression was used to evaluate the relationship for: (1) exclosure age and changes in the wetland index and (2) differences in the wetland index and width-to-depth between exclosure and grazed sites. R^2 values were used to measure the strength of that relationship using a 95% confidence level. Statistical analyses were carried out using S-PLUS (Insightful Corp, 2001).

RESULTS

Community type wetland index values were calculated for the 472 community types in the five riparian vegetation classifications (Table 2) used at the study sites. The mean CTWI for community types in all five classifications was 66, with a range from 5 to 100 (Figure 1).
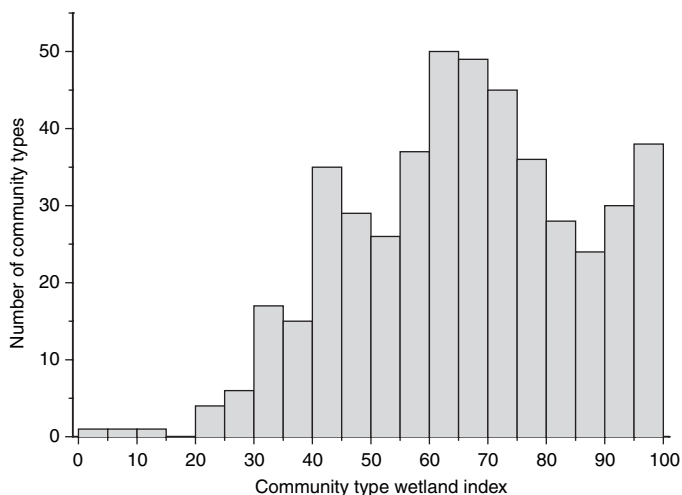

FIGURE 1. Community Type Wetland Index Values for 472 Community Types of the Five Riparian Vegetation Classifications Used in This Study.

Site wetland index values were compared for the exclosure and adjacent grazed area at 14 sites. The greenline wetland index had a mean pair-wise difference of 7.3 units between the exclosure and grazed

sites, with the average exclosure value being 11% higher than for grazed sites (Figure 2a). A one-sided paired *t*-test (95% confidence level) based on greenline data indicated that exclosure sites had a higher wetland index ($p = 0.003$) than the grazed sites (Table 4). For 11 of the 14 sites the greenline wetland index was higher in the exclosure than the grazed area.
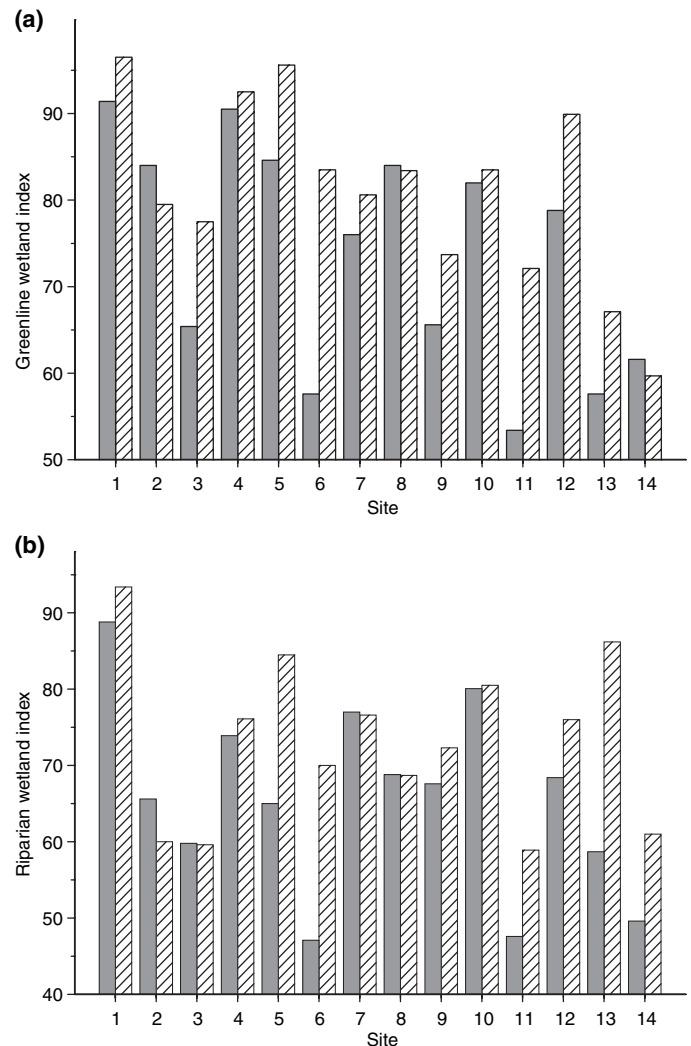

FIGURE 2. (a) Greenline and (b) Riparian Wetland Index Values for 14 Pairs of Grazed (solid fill) and Adjacent Exclosure (striped fill) in Idaho and Oregon. Sites are listed in order of increasing channel width (for exclosure channels). The lower end of the Y-axis is truncated at 40.

The riparian wetland index was also higher in the exclosure sites, by an average of 7.6 units (14%) compared with the grazed sites (Figure 2b). The one-sided paired *t*-test indicated that the exclosure sites had a higher riparian wetland index ($p = 0.006$) than the grazed sites (Table 4). For 10 of the 14 sites the riparian wetland index was higher in the exclosure than the grazed area.

TABLE 4. Results of One-sided, Paired t-Test (95% confidence level) Comparing Wetland
Index Values and Stream Channel Variables in Exclosures and Grazed Areas (n = 14 pairs).

Variable	Direction of Test[1]	Mean Difference[2]	Mean Difference (%)[3]	t-Value	p-Value
Greenline wetland index	Greater	7.33	11	3.3	0.003*
Riparian wetland index	Greater	7.56	14	2.9	0.007*
Stream gradient (percent)	Lower	−0.03	−6	−0.3	0.388
Bankfull width (m)	Lower	−0.33	−9	−1.3	0.1
Width-to-depth ratio	Lower	−3.86	−18	−3	0.005*
Entrenchment ratio	Greater	0.12	8	1.5	0.078
Bank stability (percent)	Greater	15.79	26	3.7	0.002*
Bank angle (percent)	Lower	−3.93	−4	−0.9	0.192
Undercut (percent of reach)	Greater	2.00	26	0.6	0.293

[1]One-sided t-test for difference in exclosure compared to grazed area.
[2]Exclosure minus grazed (i.e., change after livestock exclusion).
[3]Mean percent difference relative to grazed area.
*Significant at $p \leq 0.05$

In addition, linear regression was used to evaluate the relationship between exclosure age and the difference in wetland index values between grazed and exclosed areas. The R^2-values based on the linear regression of exclosure age and change in the wetland index were only 0.02 for the greenline (Figure 3a) and 0.003 for the riparian transects (Figure 3b), neither of which were statistically significant ($p = 0.64$ and $p = 0.86$, respectively). Removal of an outlier (an exclosure twice as old as the rest of the exclosures) resulted in a comparable R^2-value of 0.03 ($p = 0.57$) for the greenline, but a larger R^2-value of 0.19 ($p = 0.13$) for the riparian transects.

Bank stability was statistically higher ($p = 0.002$) in exclosures compared to adjacent grazed areas (average difference was 26%) (Table 4). Width-to-depth ratios were statistically lower ($p = 0.005$) in exclosures compared with grazed areas based on a paired t-test (average difference was −18%). Though not statistically significant, entrenchment ratio ($p = 0.078$) was slightly higher (average difference was 12%), and bankfull width ($p = 0.100$) was slightly lower (average difference was −9%) in exclosures.

There was a relatively strong negative relationship between the greenline wetland index and width-to-depth for both the exclosures ($R^2 = 0.36$; $p = 0.02$; Figure 4) and grazed sites ($R^2 = 0.29$; $p = 0.05$; Figure 5). The grazed sites had a somewhat steeper slope than the exclosure sites (correlation coefficient of −0.88 vs. −0.83). In order to evaluate the relationship between vegetation, livestock, and channel form, the difference in the wetland index between exclosures and grazed sites was compared to the difference in width-to-depth for the same pairs of exclosure and grazed sites. There was a weak relationship between differences (exclosure vs. grazed) in the wetland index and differences in width-to-depth for the greenline ($R^2 = 0.003$; Figure 6) and the riparian data ($R^2 = 0.04$).
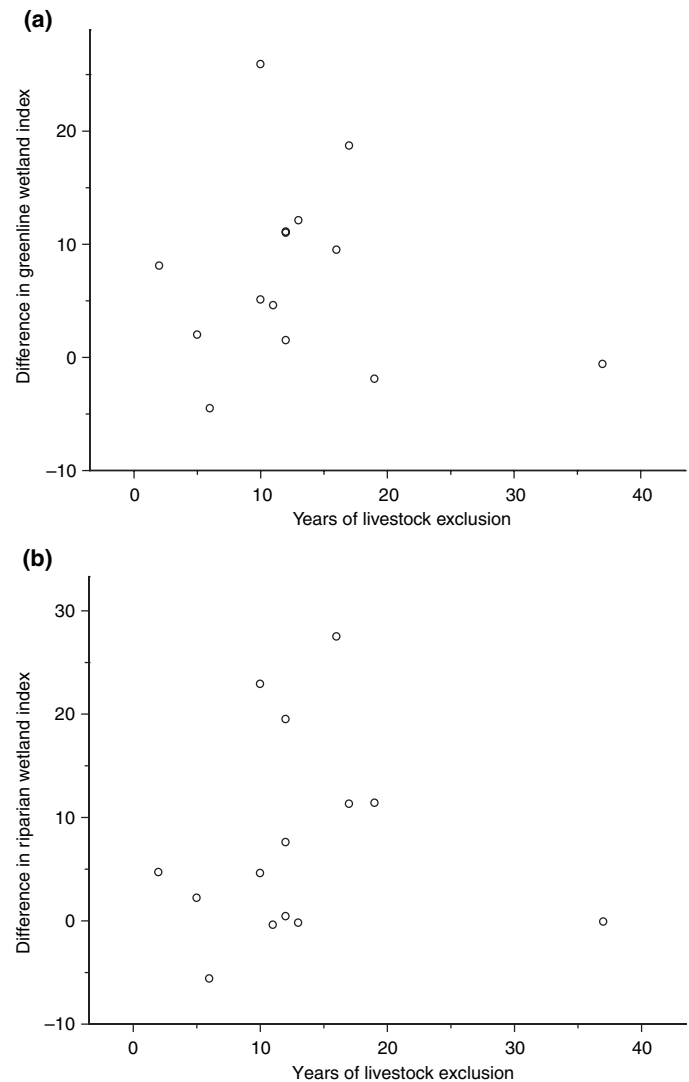

FIGURE 3. Linear Regression of Difference (exclosure minus grazed) in the Wetland Index and Time Since Livestock Exclusion for (a) Greenline and (b) Riparian Vegetation.
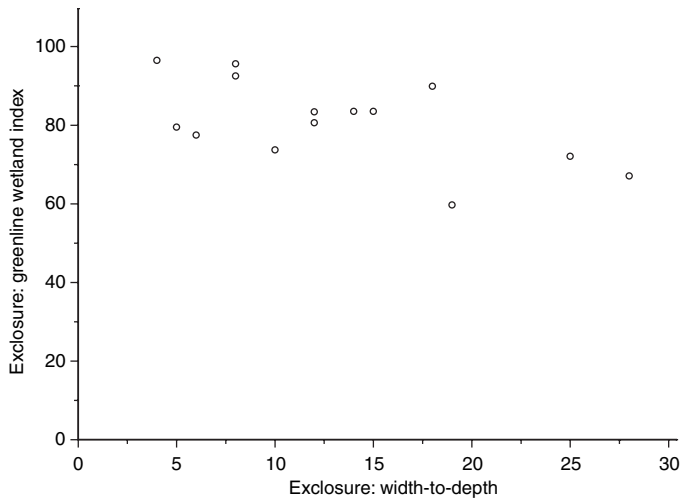
FIGURE 4. Relationship of Greenline Wetland
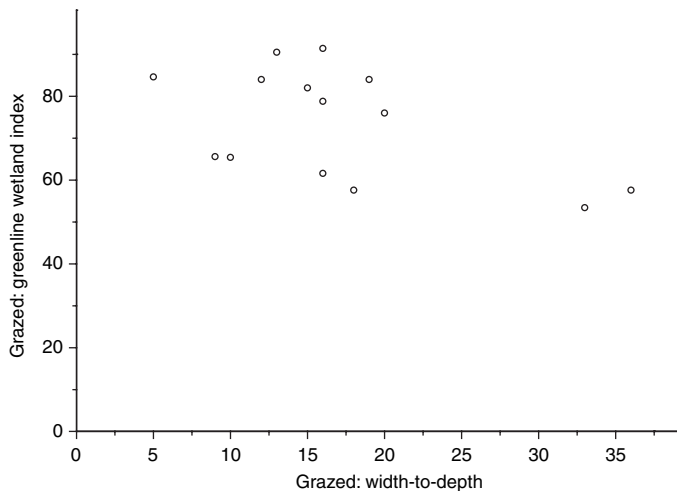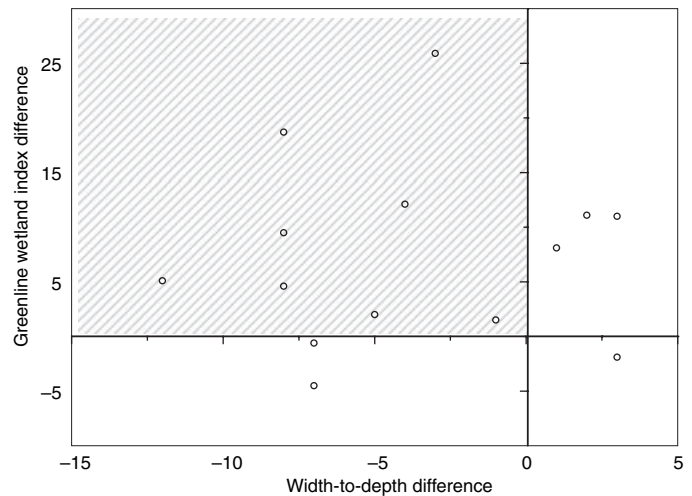Index and Width-to-Depth at Exclosure Sites.

FIGURE 6. Comparison of Differences in Greenline Wetland Index
to Differences in Width-to-Depth Between Exclosure and Grazed
Sites. Shaded area indicates exclosures that had a higher greenline
wetland index and a lower width-to-depth than adjacent grazed
sites.

FIGURE 5. Relationship of Greenline Wetland
Index and Width-to-Depth at Grazed Sites.

DISCUSSION

In semi-arid regions, livestock spend a dispropor-
tionate amount of time in riparian areas, where there
is water, forage, cooler temperatures, and relatively
flat terrain (Bryant, 1982; Kauffman and Krueger,
1984; Fleischner, 1994). As a result, livestock can
have significant impacts on riparian areas and
streams. The wetland index was developed as a tool
to measure changes in vegetation that are indicative
of changes in hydrologic processes because of
anthropogenic disturbance.

As hypothesized, sites where livestock were exclu-
ded generally had higher wetland index values, i.e., a
greater abundance of species commonly found in

wetlands. This is consistent with other studies obser-
ving changes in vegetation with livestock removal
(Beschta and Platts, 1986; National Research Coun-
cil, 2002). Schultz and Leininger (1990) noted dra-
matic increases in obligate wetland willows (*Salix*
spp.) in a 30-year-old exclosure, and a concurrent
decline in the facultative species Kentucky bluegrass
(*Poa pratensis*). Similarly, increases in obligate and
facultative wet woody species occurred on gravel bars
in 15-year-old exclosures (Green and Kauffman,
1995) and for obligate wetland sedges (primarily *Ca-
rex nebrascensis)* in a 34-year-old exclosed meadow
(Dobkin *et al.*, 1998) compared with the grazed areas
outside the exclosures that had much greater cover
of upland species (*Artemisia tridentata* and *Chrysot-
hamnus* spp.).

The possible processes that led to an increase in
obligate wetland species in the exclosures include: (1)
release from grazing pressure that may have bene-
fited wetland species more than facultative and
upland species; and (2) post-exclusion changes in
physical conditions that increased connectivity
between the stream (or ground water) and floodplain
or that increased infiltration and retention of mois-
ture in the riparian area. Release from herbivory
could partly explain the increase in wetland species
after livestock removal, particularly if wetland spe-
cies had greater palatability than facultative and
upland species as is sometimes the case [see palata-
bility ratings in Hansen *et al.* (1995); Crowe and
Clausnitzer (1997)].

Physical changes as a result of livestock can
include streambank destabilization, erosion, and soil

compaction (Kauffman *et al.*, 1983; Trimble and Mendel, 1995). Streambank disturbance and erosion can lead to channel widening or incision, and entrapment of the flow within a gully, which can reduce overbank flows and cause the riparian area to become drier (Sarr, 2002). Removal of livestock has been related to dramatic increases in infiltration; a threefold increase in wet meadows and an 11-fold increase in dry meadows (Kauffman *et al.*, 2004). Therefore, sites without livestock could capture and store much more water from precipitation and over-bank flows. This could lead to a positive feedback between moisture retention, plant growth, and soil macropore formation by roots as suggested by Thurow (1991) and Angers and Caron (1998). This is part of a larger feedback loop between hydric riparian vegetation, stable streambanks, over-bank flows, infiltration, and water storage.

Riparian species stabilize streambanks and affect channel form because of the ability of riparian species to dissipate stream energy and to trap and anchor sediment (Hughes, 1997; Naiman and Decamps, 1997). Many obligate wetland species have long, deep, and dense roots compared to species that grow in drier conditions, i.e., facultative or upland species (Manning *et al.*, 1989; Kleinfelder *et al.*, 1992; Dunaway *et al.*, 1994). Streambanks with obligate wetland sedges (*Carex* spp.) and rushes (*Juncus* spp.) showed lower susceptibility to erosion than banks with dry meadow species [silver sagebrush (*Artemisia cana*) and annual grasses] (Micheli and Kirchner, 2002). Similarly, Toledo and Kauffman (2001) detected more obligate wetland species, and twice the root biomass, along narrower, unincised reaches compared with wider, incised reaches on the same stream. Converting their data to the wetland index showed that the unincised reaches had values 30% higher than incised reaches. The abundance of obligate wetland species, quantified with the wetland index, may indirectly provide information about the ability of the riparian vegetation to stabilize streambanks, particularly for meadow streams.

Bank stability was indeed higher in the exclosures, consistent with Kauffman *et al.* (1983). Stable banks help maintain channel form and promote overbank flow (Gebhardt *et al.*, 1989), which facilitates dissipation of stream energy and infiltration and storage of water. Though not statistically significant, exclosures had slightly more undercuts and slightly steeper banks (higher bank angle), which is consistent with greater undercuts and less bank erosion associated with livestock exclusion reported by Kauffman *et al.* (1983).

While livestock exclusion was associated with an increase in the wetland index, there was no significant relationship between time since exclusion and the wetland index. When an outlier (right-most point in Figure 3b) was removed, a slightly stronger linear relationship was observed between the riparian wetland index and time since exclusion These data suggest that some sites respond more quickly than others, which is consistent with the concept of variable recovery trajectories after livestock exclusion by Sarr (2002).

The process of channel narrowing is one of the most consistent adjustments after livestock exclusion (Magilligan and McDowell, 1997) and has been observed in multiple studies (Platts, 1981; Hubert *et al.*, 1985; Myers and Swanson, 1995; Clary, 1999). In our study, the average channel width in exclosures was less than that of adjacent grazed areas, although the difference was not statistically significant. Possible reasons for the lack of significantly narrower channels in exclosures are: (1) some sites may have had minimal channel widening prior to exclusion; (2) the legacy of excessive disturbance prior to exclusion, which could have created feedback loops (entrapment, impoverishment, and convergence) that kept stream/riparian systems in a degraded state (Sarr, 2002); (3) observer variability which, although relatively small for these channel and bank measurements (Roper *et al.*, 2002; Archer *et al.*, 2004), could have interfered with detection of actual change; or (4) the slow process of channel narrowing compared with vegetation change (Kondolf, 1993). These points, particularly the latter, could explain why channel narrowing after livestock exclusion was small and not statistically significant in this data set and in studies by Kondolf (1993) and Allen-Diaz *et al.* (1998).

Streams within exclosures did have statistically smaller width-to-depth ratios. To evaluate the relationship of channel narrowing and/or deepening with changes in hydric vegetation along the stream, we compared differences (exclosure *vs.* grazed) in the wetland index to differences in the width-to-depth ratio. There was no linear relationship, as indicated by the R^2-values below 0.04. However, we did observe a general pattern, with the largest differences observed in exclosures that had a combination of greater wetland index values and smaller width-to-depth ratios (upper left quadrant of Figure 6). This pattern did not occur in the other six pairs of exclosure and grazed sites, although the deviations from the pattern were relatively small, and all but one exclosure had either a greater wetland index or a lower width-to-depth ratio. While these data do not conclusively show that channel narrowing and/or deepening is associated with increased hydric vegetation along the stream, the hypothesis remains plausible based on these data.

Three limitations of this study were the generality of community types, subjectivity of the WIS, and

inherent problems with exclosure studies. In the initial years of PIBO monitoring, which are the source of the data evaluated here, community types were used to describe riparian vegetation based on the recently published protocol by Winward (2000). Since community types are generalizations, they do not always accurately represent the site vegetation. This imprecise fit between vegetation at a site and the community type description in the classification contributes to observer variability; a test with multiple technicians found only 39% average agreement of the community type recorded over 1-m increments (Coles-Ritchie et al., 2004). However, not all of this variability is ecologically important, as where observers record different but ecologically equivalent community types. That explains, at least in part, why tests showed relatively low observer variability in the wetland index generated from community type data (Coles-Ritchie et al., 2004).

Another problem with community types was that classifications had not been performed for all geographical areas, hence sometimes community types from distant areas were the only option for describing site vegetation. In addition, there were limited ways to analyze community type data from multiple classifications, which was the reason a CTWI was developed in this study. The CTWI proved useful, but calculating it for community types was a laborious process. Because of these problems with community types, PIBO has switched to species data collection for its riparian monitoring.

The WIS is a subjective categorization of species based on observation of the habitats where species are most commonly found. As a result, there can be debate about the appropriateness of assignments for certain species. When the WIS for multiple species at a site are used, as is performed with the wetland index, a pattern should emerge that accurately represents the site. While community types do not perfectly represent the vegetation at a given step or series of steps, they generally represented the types of species (in terms of the WIS) that were present. Therefore, the wetland index for community types seemed to be a satisfactory, although imprecise, representation of the dominant vegetation at the sites.

There are advantages and disadvantages to using exclosures to evaluate the influence of livestock (Sarr, 2002). A major advantage is that environmental site variability (precipitation, geology, flow regime, etc.) is practically eliminated. Therefore, the influence of livestock (or livestock removal) can be isolated, while other variables remain constant. Limitations of livestock exclosures include their small size (relative to the larger watershed which influences runoff and sediment yield), haphazard placement, and variable

or unknown site histories in terms of livestock activity, which can lead to inconsistent results (Sarr, 2002). Another limitation of many riparian exclosure studies is that the data are from sites on the same stream (Kauffman et al., 1983; Schultz and Leininger, 1990; Allen and Marlow, 1994; Green and Kauffman, 1995; Case and Kauffman, 1997; Dobkin et al., 1998; Clary, 1999; Brookshire et al., 2002) or from adjacent watersheds (Leege et al., 1981), leading to small sample sizes, questions of pseudoreplication, and an inability to extrapolate results to other areas.

Our study compared riparian vegetation in and outside exclosures at 14 locations in two states. This type of broad-scale analysis eliminated some of the limitations associated with other exclosure studies. First, only one site was evaluated within each watershed, which eliminated concerns of pseudo-replication that exist for studies using multiple exclosures on the same stream (Rinne and LaFayette, 1991). Secondly, the relatively large sample size allowed us to make broader generalizations about response of vegetation, and the wetland index, as a result of livestock exclusion. The importance of sample size in detecting change is highlighted by Archer et al. (2004) and Roper et al., 2002 for physical stream attributes and by Coles-Ritchie et al. (2004) for riparian vegetation variables.

The comparable environmental conditions between the exclosure and adjacent grazed area isolated the influence of livestock, which likely enhanced our ability to detect differences in the wetland index. Where studies compare sites from different geographical areas, site variability (precipitation, geology, flow regime, etc.) needs to be accounted for (as done by Milchunas and Lauenroth, 1993; Kershner et al., 2004b). Also, our study sites occurred in relatively dry regions, where there are distinct differences in vegetation between riparian areas and uplands. In wetter areas, where there is greater similarity between vegetation in riparian and upland settings, the responsiveness and usefulness of the wetland index would need to be evaluated.

CONCLUSION

The wetland index provides a tool for characterizing riparian vegetation and evaluating how disturbance affects processes that influence riparian vegetation. An increase in the wetland index is indicative of a shift to species that are more frequently found in wetlands, which likely indicates greater connectivity between the stream (or ground water) and the floodplain. Conversely, a decrease in the wetland

index likely indicates less stream/floodplain connectivity, which can have negative implications for the stream channel (especially fish habitat) and the riparian ecosystem.

The wetland index has a number of strengths as a tool for summarizing riparian vegetatation data. It can be used to compare sites with similar environmental conditions, as presented here, or to monitor the same sites over time. It can also be applied consistently across a large geographical area, since a WIS has been assigned to most species occurring in wetlands across the United States (Reed, 1996).

In this study, the wetland index was used to compare riparian exclosures and adjacent grazed sites, based on community type data, demonstrating one application of the wetland index. There are a number of limitations with using community types for riparian monitoring (Coles-Ritchie *et al.*, 2004), which can be avoided by collecting species data, as is now being done by PIBO. Nothwistanding the imprecise nature of community types, the wetland index (based on community types) permitted detection of consistent differences between exclosures and adjacent grazed areas, with the exclosures having higher wetland index values. This indicates that vegetation, and the wetland index in particular, was responsive to livestock removal.

LITERATURE CITED

Allen, D.R. and C.B. Marlow, 1994. Shoot Population Dynamics of Beaked Sedge Following Cattle Grazing. *Journal of Range Management* 47:64-69.

Allen-Diaz, B., R.D. Jackson, and J.S. Fehmi, 1998. Detecting Channel Morphology Change in California's Hardwood Rangeland Spring Ecosystems. *Journal of Range Management* 51:514-518.

Angers, D.A. and J. Caron, 1998. Plant-Induced Changes in Soil Structure: Processes and Feedbacks. *Biogeochemistry* 42:55-72.

Archer, E.K., B.B. Roper, R.C. Henderson, J.L. Kershner, and S.C. Mellison, 2004. *Testing Common Stream Sampling Methods: How Useful are These Techniques for Broad-Scale, Long-Term Monitoring?*. USDA Forest Service RMRS-GTR-122, Ft. Collins, Colorado.

Bauer, S.B. and T.A. Burton, 1993. *Monitoring Protocols to Evaluate Water Quality Effects of Grazing Management on Western Rangeland Streams*. Environmental Protection Agency EPA-910/R-9-93-017, Seattle, Washington.

Belsky, A.J., A. Matzke, and S. Uselman, 1999. Survey of Livestock Influences on Stream and Riparian Ecosystems in the Western United States. *Journal of Soil and Water Conservation* 54(1):419-431.

Beschta, R.L. and W.S. Platts, 1986. Morphological Features of Small Streams: Significance and Function. *Water Resources Bulletin* 22(3):369-379.

Bohn, C.C. and J.C. Buckhouse, 1985. Some Responses of Riparian Soils to Grazing Management in Northeastern Oregon. *Journal of Range Management* 38(4):378-381.

Brookshire, E.N.J., J.B. Kauffman, D. Lytjen, and N. Otting, 2002. Cumulative Effects of Wild Ungulate and Livestock Herbivory on Riparian Willows. *Oecologia* 132:559-566.

Brown, S.C., 1999. Vegetation Similarity and Avifaunal Food Value of Restored and Natural Marshes in Northern New York. *Restoration Ecology* 7(1):56-68.

Bryant, L.D., 1982. Response of Livestock to Riparian Zone Exclusion. *Journal of Range Management* 35(6):780-785.

Case, R.L. and J.B. Kauffman, 1997. Wild Ungulate Influences on the Recovery of Willows, Black Cottonwood and Thinleaf Alder Following Cessation of Cattle Grazing in Northeastern Oregon. *Northwest Science* 71(2):115-126.

Castelli, R.M., J.C. Chambers, and R.J. Tausch, 2000. Soil-Plant Relations Along a Soil-Water Gradient in Great Basin Riparian Meadows. *Wetlands* 20:251-266.

Chapin, D.M., R.L. Beschta, and H.W. Shen, 2002. Relationships Between Flood Frequencies and Riparian Plant Communities in the Upper Klamath Basin, Oregon. *Journal of the American Water Resources Association* 38(3):603-617.

Clary, W.P., 1999. Stream Channel and Vegetation Responses to Late Spring Cattle Grazing. *Journal of Range Management* 52(3):218-227.

Coles-Ritchie, M.C., 2005. Evaluation of Riparian Vegetation Data and Associated Sampling Techniques. PhD Dissertation, Utah State University, Logan, Utah.

Coles-Ritchie, M.C., R.C. Henderson, E.K. Archer, C. Kennedy, and J.L. Kershner, 2004. *Repeatability of Riparian Vegetation Sampling Methods: How Useful Are These Techniques for Broad-scale, Long-Term Monitoring*. USDA Forest Service RMRS-GTR-138, Ft. Collins, Colorado.

Crowe, E.A. and R.R. Clausnitzer, 1997. *Mid-Montane Wetland Plant Associations of the Malheur, Umatilla and Wallowa-Whitman National Forests*. USDA Forest Service R6-NR-ECOL-TP-22-97, Portland, Oregon.

Dobkin, D.S., A.C. Rich, and W.H. Pyle, 1998. Habitat and Avifaunal Recovery from Livestock Grazing in a Riparian Meadow System of the Northwestern Great Basin. *Conservation Biology* 12(1):209-221.

Dunaway, D., S. Swanson, J. Wendel, and W. Clary, 1994. The Effect of Herbaceous Plant Communities and Soil Textures on Particle Erosion of Alluvial Streambanks. *Geomorphology* 9:47-56.

Dwire, K.A., J.B. Kauffman, E.N. Brookshire, and J.E. Baham, 2004. Plant Biomass and Species Composition Along an Environmental Gradient in Montane Riparian Meadows. *Oecologia* 139:309-317.

Elmore, W. and J.B. Kauffman, 1994. Riparian and Watershed Systems: Degradation and Restoration. In: *Ecological Implications of Livestock Herbivory*, M. Vavra, W.A. Laycock, and R.D. Pieper (Editors). Western Society of Range Management, Denver, Colorado, pp. 213-231.

Fitch, L. and R.L. Adams, 1998. Can Cows and Fish Co-Exist? *Canadian Journal of Plant Science* 78(2):191-198.

Fleischner, T.L., 1994. Ecological Costs of Livestock Grazing in Western North America. *Conservation Biology* 8(3):629-644.

Gebhardt, K.A., C. Bohn, S. Jensen, and W. Platts, 1989. Use of Hydrology in Riparian Classifications. In: *Practical Approaches to Riparian Resource Management: An Educational Workshop*, R.E. Gresswell, B.A. Barton, and J.L. Kershner (Editors). USDI Bureau of Land Management, Billings, Montana, pp. 53-59.

Green, D.M. and J.B. Kauffman, 1995. Succession and Livestock Grazing in a Northeastern Oregon Riparian Ecosystem. *Journal of Range Management* 48:307-313.

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins, 1991. An Ecosystem Perspective of Riparian Zones: Focus on Links Between Land and Water. *BioScience* 41(8):540-551.

Gurnell, A., 1997. The Hydrological and Geomorphological Significance of Forested Floodplains. *Global Ecology and Biogeography Letters* 6:219-229.

Hansen, P.L., R.D. Pfister, K. Boggs, B.J. Cook, J. Joy, and D.K. Hinckley, 1995. *Classification and Management of Montana's Riparian and Wetland Sites*. University of Montana School of Forestry, Missoula, Montana.

Hubert, W.A., R.P. Lanka, T.A. Wesche and F. Stabler, 1985. Grazing Management Influences on Two Brook Trout Streams in Wyoming. In: *Riparian Ecosystems and Their Management: Reconciling Conflicting Uses*, R.R. Johnson, C.D. Ziebell, D.R. Patton, P.F. Ffolliott, and R.H. Hamre (Editors). USDA Forest Service RMRS-GTR-120, Fort Collins, Colorado, pp. 290-294.

Hughes, F.M.R., 1997. Floodplain Biogeomorphology. *Progress in Physical Geography* 21(4):501-529.

INFISH (Inland Fish Strategy), USDA Forest Service, 1995. *Inland Native Fish Strategy. Decision Notice/Finding of No Significant Impact. Environmental Assessment, Inland Native Fish Strategy, Interim Strategies for Managing Fish-Producing Watersheds in Eastern Oregon and Washington, Idaho, Western Montana, and Portions of Nevada*. USDA Forest Service, Washington, D.C.

Insightful Corp., 2001. *S-Plus 6.0 Professional*. Lucent Technologies Inc., Seattle, Washington.

Kauffman, J.B. and W.C. Krueger, 1984. Livestock Impacts on Riparian Ecosystems and Streamside Management Implications…A Review. *Journal of Range Management* 37(5):430-438.

Kauffman, J.B., W.C. Krueger, and M. Varva, 1983. Impacts of Cattle on Streambanks in Northeastern Oregon. *Journal of Range Management* 36(6):683-685.

Kauffman, J.B., A.S. Thorpe, and E.N.J. Brookshire, 2004. Livestock Exclusion and Belowground Ecosystem Responses in Riparian Meadows of Eastern Oregon. *Ecological Applications* 14(6):1671-1679.

Kershner, J.L., E.K. Archer, M. Coles-Ritchie, E. Cowley, R.C. Henderson, K. Kratz, C. Quimby, D.M. Turner, L.C. Ulmer, and M.R. Vinson, 2004a. *Guide to Effective Monitoring of Aquatic and Riparian Resources*. USDA Forest Service RMRS-GTR-121, Ft. Collins, Colorado.

Kershner, J.L., B.B. Roper, N. Bouwes, R. Henderson, and E. Archer, 2004b. An Analysis of Stream Habitat Conditions in Reference and Managed Watersheds on Some Federal Lands Within the Columbia River Basin. *North American Journal of Fisheries Management* 24:1363-1375.

Kleinfelder, D., S. Swanson, G. Norris, and W. Clary, 1992. Unconfined Compressive Strength of Some Streambank Soils With Herbaceous Roots. *Soil Science Society of America Journal* 56:1920-1925.

Kondolf, G.M., 1993. Lag in Stream Channel Adjustment to Livestock Exclosure, White Mountains, California. *Restoration Ecology* 1:226-230.

Kovalchik, B.L., 1987. *Riparian Zone Associations: Deschutes, Ochoco, Fremont, and Winema National Forests*. USDA Forest Service Technical Paper 279-87, Portland, Oregon.

Leege, T.A., D.J. Herman, and B. Zamora, 1981. Effects of Cattle Grazing on Mountain Meadows in Idaho. *Journal of Range Management* 34:324-328.

Lohman, K., 2004. Wildlife Use of Riverine Wetland Habitats. In: *Wetland and Riparian Areas of the Intermountain West: Ecology and Management*, M.C. McKinstry, W.A. Hubert, and S.H. Anderson (Editors). University of Texas Press, Austin, Texas, pp. 74-86.

Magilligan, F.J. and P.F. McDowell, 1997. Stream Channel Adjustments Following Elimination of Cattle Grazing. *Journal of the American Water Resources Association* 33(4):867-878.

Manning, M.E. and W.G. Padgett, 1995. *Riparian Community Type Classification for Humboldt and Toiyabe National Forests, Nevada and Eastern California*. USDA Forest Service R4-ECOL-95-01, Ogden, Utah.

Manning, M.E, S.R. Swanson, T. Svejcar, and J. Trent, 1989. Rooting Characteristics of Four Intermountain Meadow Community Types. *Journal of Range Management* 42(4):309-312.

Micheli, E.R. and J.W. Kirchner, 2002. Effects of Wet Meadow Riparian Vegetation on Streambank Erosion. 1. Remote Sensing Measurements of Streambank Migration and Erodibility. *Earth Surface Processes and Landforms* 27:627-639.

Michener, M.C., 1983. Wetland Site Index for Summarizing Botanical Studies. *Wetlands* 3:180-191.

Milchunas, D.G. and W.K. Lauenroth, 1993. Quantitative Effects of Grazing on Vegetation and Soils Over a Global Range of Environments. *Ecological Monographs* 63:327-366.

Montgomery, D.R. and L.H. McDonald, 2002. Diagnostic Approach to Stream Channel Assessment and Monitoring. *Journal of the American Water Resources Association* 38:1-16.

Myers, T.J. and S. Swanson, 1995. Impact of Deferred Rotation Grazing on Stream Characteristics in Central Nevada: A Case Study. *North American Journal of Fisheries Management* 15:428-439.

Naiman, R.J. and H. Decamps, 1997. The Ecology of Interfaces: Riparian Zones. *Annual Review of Ecology and Systematics* 28:621-658.

National Research Council, 2002. *Riparian Areas: Functions and Strategies for Management*. National Academy Press, Washington, D.C.

Overton, C.K., S.P. Wollrab, B.C. Roberts, and M.A. Radko, 1997. *R1/R4 (Northern and Intermountain Regions) Fish and Fish Habitat Standard Inventory Procedures Handbook*. USDA Forest Service INT-GTR-346, Ogden, Utah.

PACFISH (Pacific Anadromous Fisheries Habitat), USDA Forest Service, USDI Bureau of Land Management, 1994. *Environmental Assessment for the Implementation of Interim Strategies for Managing Anadromous Fish-Producing Watersheds in Eastern Oregon, Washington, Idaho, and Portions of California*. USDA Forest Service, Washington, D.C.

Padgett, W.G., A.P. Youngblood, and A.H. Winward, 1989. *Riparian Community Type Classification of Utah and Southeastern Idaho*. USDA Forest Service R4-ECOL-89-01, Ogden, Utah.

Platts, W.S., 1981. Sheep and Streams. *Rangelands* 3(4):158-160.

Platts, W.S., 1991. Livestock Grazing. In: *Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats*, W.R. Meehan (Editor). American Fisheries Society Special Publication 19, Bethesda, Maryland, pp. 389-423.

Reed, P.B. Jr, 1996. *Draft Revision of 1988 National List of Plant Species That Occur in Wetlands: National Summary*. U.S. Fish and Wildlife Service, Washington, D.C.

Rinne, J.N. and R.A. LaFayette, 1991. *Southwestern Riparian-Stream Ecosystems: Research Design, Complexity, and Opportunity*. USDA Forest Service RM-299, Ft. Collins, Colorado.

Roper, B.B., J.L. Kershner, E.K. Archer, R.C. Henderson, and N. Bouwes, 2002. An Evaluation of Physical Habitat Attributes Used to Monitor Streams. *Journal of the American Water Resources Association* 38:1637-1646.

Saab, V.A., C.E. Bock, T.E. Rich, and D.S. Dobkin, 1995. Livestock Grazing Effects in Western North America. In: *Ecology and Management of Neotropical Migratory Birds: A Synthesis and Review of Critical Issues*, T.E. Martin and D.M. Finch (Editors). Oxford University Press, New York, pp. 311-353.

Sarr, D.A., 2002. Riparian Livestock Exclosure Research in the Western United States: A Critique and Some Recommendations. *Environmental Management* 30(4):516-526.

Schultz, T.T. and W.C. Leininger, 1990. Differences in Riparian Vegetation Structure Between Grazed and Exclosures. *Journal of Range Management* 43(4):295-299.

Stromberg, J.C., 2001. Biotic Integrity of Platanus Wrightii Riparian Forests in Arizona: First Approximation. *Forest Ecology and Management* 142:251-266.

Tabacchi, E., D.L. Correll, R. Hauer, G. Pinay, A. Planty-Tabacchi, and R.C. Wissmar, 1998. Development, Maintenance and Role of Riparian Vegetation in the River Landscape. *Freshwater Biology* 40:497-516.

Tabacchi, E., L. Lambs, H. Guilloy, A. Planty-Tabacchi, E. Muller, and H. Decamps, 2000. Impacts of Riparian Vegetation on Hydrological Processes. *Hydrological Processes* 14:2959-2976.

Tart, D., K. Brewer, B. Schwind, C. Williams, R. Warbington, J. DiBenedetto, E. Crowe, W. Clerke, M. Girard, H. Gordon, K. Sleavin, P. Krosse, M. Manning, L. Suring, M. Schanta, J. Haglund, B. Short, and D. Wheeler, 2002. *Existing Vegetation Classification and Mapping Technical Guide – REVIEW DRAFT*. USDA Forest Service, Washington, D.C.

Thurow, T.L., 1991. Hydrology and Erosion. In: *Grazing Management: An Ecological Perspective*, R.K. Heitschmidt and J.W. Stuth (Editors). Timber Press, Portland, Oregon, pp. 141-159.

Toledo, Z.O. and J.B. Kauffman, 2001. Root Biomass in Relation to Channel Morphology of Headwater Streams. *Journal of the American Water Resources Association* 37(6):1653-1663.

Trimble, S.W. and A.C. Mendel, 1995. The Cow as a Geomorphic Agent – A Critical Review. *Geomorphology* 13:233-253.

USDIFWS (U.S. Department of Interior, Fish and Wildlife Service), 1998. Endangered and Threatened Wildlife and Plants: Determination of Threatened Status for the Klamath River and Columbia River Distinct Population Segments of Bull Trout Final Rule. Federal Register June 10, 1998 (Volume 63, Number 11, Pages 31647-31674) 50 CFR Part 17, RIN 1018-AB94.

Wakeley, J.S. and R.W. Lichvar, 1997. Disagreements Between Plot-Based Prevalence Indices and Dominance Ratios in Evaluations of Wetland Vegetation. *Wetlands* 17(2):301-309.

Wentworth, T.R., G.P. Johnson, and R.L. Kologiski, 1988. Designation of Wetlands by Weighted Averages of Vegetation Data: A Preliminary Evaluation. *Water Resources Bulletin* 24:389-396.

Wheeler, M.A., M.J. Trlica, G.W. Frasier, and J.D. Reeder, 2002. Seasonal Grazing Affects Soil Physical Properties of a Montane Riparian Community. *Journal of Range Management* 55:49-56.

Winward, A.H., 2000. *Monitoring the Vegetation Resources in Riparian Areas*. USDA Forest Service RMRS-GTR-47, Ogden, Utah.